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## Estimating the Benefits of Reductions in Low-Level Environmental Risks

by

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IES Staff Paper No. 26  
March, 1985



institute for environmental studies

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## Abstract

This paper examines the potential for using the results of studies of wage-risk premiums for hazardous jobs to estimate the benefits of reductions in low-level environmental risks. An economic model of risk reduction is developed that intuitively relates the benefits of reductions in environmental risks to risk reduction in the workplace. Inherent biases in the wage-risk premiums are considered and shown to be downward biased estimates of benefits. We conclude with an example that considers the potential benefits of not allowing a partial conversion of the U.S. light-duty vehicle fleet to diesel-powered engines.



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## I: Introduction

There has been a growing trend since the mid 1970s toward the requirement of comprehensive analyses of the economic impacts of proposed government regulations. This trend is evidenced most recently by the issuance of Executive Order 12,291 (E.O. 12,291) in 1981. The primary thrust of E.O. 12,291 is that federal administrative agencies conduct a benefit-cost analysis of all proposed and existing major regulations to evaluate the net social benefits that those regulations can be expected to generate.

While the goal of Executive Order 12,291 is certainly commendable from the perspective of efficiency, it ignores the problem of attaching a dollar measure to the benefits from a proposed regulation that are non-market in nature. We are concerned with how one might attach a preliminary value to the benefits from a regulation that reduces the level of a given risk to human life. The focus is on low-level environmental risks; a subject of much of the U.S. Environmental Protection Agency's regulatory activities.

This paper examines the potential for using results from studies of the value of wage-risk premiums in the labor market to establish a lower bound on the monetary benefits realized from reducing a particular low-level environmental risk.<sup>1</sup> The use of these estimates could greatly facilitate the timely evaluation of alternative policies designed to mitigate such risks while adhering to the spirit of the requirements of E.O. 12,291. For example, these estimates could be used to select a subset of the most promising alternatives to a given problem that could then be analyzed in more rigorous detail. Additionally, if the estimates considered here are downward biased for a given reduction in environmental risks and costs are accurately measured, then policies legitimized via a cost-benefit







analysis utilizing these figures would have a reasonably high probability of being correct.

Part II begins with a discussion of the nature of low-level environmental risks. A simple model is then developed that identifies the equilibrium condition for achieving an efficient reduction in a given low-level environmental risk and the applicability of the estimates considered in Part III is discussed. Part III considers recent efforts to place a value on the wage premium required for an individual to accept a specific probability of death. In Part IV the approach to benefits estimation outlined here is applied to a specific situation. Part V summarizes the analysis and conclusions are offered.

## II: The Nature of Low-Level Environmental Risks

From the policymaker's perspective low-level environmental risks—those associated with the landfill disposal of hazardous wastes, the release of toxics from an industrial site, pesticides, etc.—pose a particularly difficult problem. In the absence of specific guidelines, it is not clear what objective should be pursued vis-a-vis reducing such risks. One alternative is to pursue policies that will reduce those risks below a pre-determined threshold level. Risk levels falling below this "De Minimis" level are assumed to have a minimal impact on society's general level of well-being. From the economist's perspective, this "De Minimis" approach to risk reduction is flawed because it ignores the fundamental rule of economic efficiency, i.e., actions should only be pursued up to the point where the marginal benefits realized equal the marginal costs incurred. Additionally, such an approach ignores the requirements of E.O. 12,291.



However, the notion of attempting to achieve an economically efficient level of risk reduction in the context of environmental risks has been assailed as being impractical if not simply impossible to achieve (Office of Technology Assessment, 1983).

The primary problem cited concerns the estimation of the monetary benefits realized when a particular risk is reduced by some amount. This problem stems from two factors. The first factor concerns the technical difficulties encountered in attempts to measure the level of a specific risk, i.e., risk assessment. As an example, it is presently quite difficult to establish the risk associated with disposing of a particular hazardous waste stream in landfills (Fisher, 1983). The second factor is the general unavailability of precise estimates of the dollar value that individuals attach to reductions in low-level environmental risks.

Quantitatively, low-level environmental risks can be conveniently characterized as the probability of certain adverse effects occurring, e.g., mortality, morbidity, and environmental damage as a result of the presence of some toxic substance in the environment (Carlson, 1984; Smith and Desvousges, 1983b). In general, the probability of realizing some adverse effect from a given environmental risk is the product of a series of temporally arranged "intermediate" probabilities. Consider the following example. The risk posed by the landfill disposal of hazardous wastes can be characterized as a function of a series of events, each occurring with probability  $P(i)$  where  $i$  refers to the event in question. In order of occurrence the events are failure, contamination, exposure, and the receiving of a dose of the toxic substance in question in excess of some threshold level necessary to induce an adverse effect. These events interact to



result in one or a combination of different outcomes including mortality, morbidity, and environmental damage. The probability of, say, mortality can be written as:

$$(1) \quad P(M) = P(F) \cdot P(C|F) \cdot P(E|C,F) \cdot P(T|E,C,F)$$

where M is mortality, F is failure, C is contamination, E is exposure, and T is exposure in excess of the threshold level necessary to induce mortality. We can further distinguish between the probability of mortality (or any adverse effect) in the near term, say one year, and the probability of mortality more than one year from now (Carlson, 1984). This recognizes the fact that many environmental hazards pose a substantial threat to both current and future generations.<sup>2</sup>

Inspection of Equation (1) suggests that in many instances the probability of some adverse occurrence may be quite small. Additionally, the estimated value of each of the intermediate probabilities possesses an associated degree of error, increasing the degree of error associated with the ultimate probability of interest. These problems directly affect the estimated monetary benefits of any reduction in risk.

The task then of assessing the monetary benefits realized from reductions in low-level environmental risks is a formidable one. However, a considerable amount of work has been undertaken to address both facets of the problem just described. In the case of risk assessment, for example, the U.S. EPA has sponsored research to develop methods for assessing the risks associated with the use of various methods for the disposal of hazardous wastes (ICF, Inc., 1984). With respect to the valuation of reductions in



those risks, Smith and Desvousges (1983a, 1983b) and Burness, et al. (1983) have provided analyses of the theoretical and empirical aspects of individuals' willingness to pay. The results of these studies suggest that with additional work it will be possible to gain meaningful insights into individuals' valuations of reductions in a variety of low-level environmental risks.

### Modeling the Benefits and Costs of Risk Reduction

Where society determines that an environmental risk is unacceptable the question is how much should the risk be reduced. With respect to economic efficiency, that question is answered by examining the marginal benefits and costs of risk reduction. Owing to the indivisible and non-excludable character of reductions in low-level environmental risks resulting from specific policies these reductions constitute a public good. From the policymaker's perspective the marginal benefits of a given reduction in risk are therefore equal to the sum of the affected individuals' marginal willingness to pay for the reduction in risk realized. In order to achieve an efficient solution, risk reduction should be pursued up to the point where the marginal benefits summed just equal the marginal costs incurred. It has been shown elsewhere that under a reasonable set of assumptions the reduction in risk will entail increased production costs for the firms generating those risks and consequently the costs of risk reduction are equal to the lost consumer and producer surplus in that market (Carlson, Johnson, and Ulen, 1984).

Most models of utility maximization where risk is a variable make use of the expected utility framework. However, use of the expected utility function has recently come into question based upon empirical observations





of how individuals behave in certain risk situations. For a discussion of the related issues, the reader is referred to Schoemaker (1982) and Weinstein and Quinn (1983). The upshot of this controversy is that in what follows we treat risk reduction as simply another argument in the individual's utility function. In a sense then, risk reduction is something the individual can "purchase." The "price" of risk reduction can be interpreted as the value of other goods and services (both private and public) foregone or wage premiums foregone.<sup>3</sup>

We model the benefits and costs of risk reduction as follows. With respect to benefits, assume that an individual is faced with the following utility maximization problem:

$$(2) \quad \begin{aligned} \text{Max } U &= U(Q, K, \bar{P}(M)) \\ \text{s.t. } Y &= p_Q Q + p_K K + p_P \bar{P}(M) \end{aligned}$$

where  $Q$  is a good, production of which results in some low-level environmental risk,  $K$  is a good not associated with such risks,  $\bar{P}(M)$  is one minus the probability of mortality associated with the risk in question,  $Y$  is income, and  $p_Q$ ,  $p_K$ , and  $p_P$  are the prices of  $Q$ ,  $K$ , and  $\bar{P}(M)$ .<sup>4</sup> Combining first order conditions and solving for  $p_P$  in Equation (2) yields the following solution:

$$(3) \quad p_P = p_Q \cdot \frac{U_{\bar{P}(M)}}{U_Q}$$

where subscripts denote partial derivatives. The right hand expression in Equation (3) measures the individual's marginal willingness to pay for a decrease in the probability of mortality. Additionally, note that



$\frac{U_{\bar{P}(M)}}{U_Q}$  is simply the marginal rate of substitution between risk reduction and good Q. The right side of Equation (3) is therefore the cost, in terms of the amount of Q given up, of getting one more unit of risk reduction.

Since a reduction in a low-level environmental risk is a public good, the total benefits of an incremental reduction<sup>5</sup> of a given risk are found by summing the marginal benefits realized across all of the affected individuals ( $i = 1, \dots, n$ ), or

$$(4) \quad \sum_{i=1}^n p_Q \cdot \frac{U_{\bar{P}(M)_i}}{U_{Q_i}} = \text{Total Benefits.}$$

In a competitive market, the costs of risk reduction, as noted above, are equal to the lost consumer and producer surplus incurred in the market where the risk is generated. This is based on the assumption that in order to reduce the risk in question, firms in the affected market will incur increased production costs, resulting in an upward shift of the aggregate supply curve. Let the inverse supply curve for a given product, production of which generates an environmental risk be characterized as:

$$(5) \quad S = S(Q, W, V, \dots)$$

where Q is the quantity of the good, W is a vector of prices of inputs to the production process, and V is technology. If a policy is then implemented that forces producers to internalize all or part of the risk they are generating, the result will be a new supply curve,  $S^*$ : Assuming that firms in the affected market operate in a cost minimizing manner,  $S^*$  will lie either on top of, or to the left of S.



The inverse aggregate demand function for the good in question is assumed to exist and be of the form:

$$(6) \quad P = P(Q, Y, K, \bar{P}(M),)$$

where  $P$  is price and  $Y$  is income. Combining Equations (5) and (6), the welfare loss, i.e., loss in producer and consumer surplus, associated with a policy that forces firms to internalize all or part of the risk in question is then equal to:

$$(7) \quad \int_0^{Q_1} [P(Q, Y, K, \bar{P}(M)) - S(Q, W, V, \dots)] dQ \\ - \int_0^{Q_2} [P(Q, Y, K, \bar{P}(M)) - S^*(Q, W, V, \dots)] dQ \geq 0$$

where  $Q_1$  and  $Q_2$  are the pre- and post-policy equilibrium levels of output respectively. Assuming that the policy results in a reduction in risk of one "unit", then Equation (7) represents the marginal social cost of that reduction in risk.

Finally, the equilibrium level of risk reduction is where Equation (4) is equal to Equation (7) or,

$$(8) \quad \sum_{i=1}^n P_{Q_i} \cdot \frac{U_{\bar{P}(M)_i}}{U_{Q_i}} = \int_0^{Q_1} [P(Q, Y, K, \bar{P}(M)) - S(Q, W, V, \dots)] dQ \\ - \int_0^{Q_2} [P(Q, Y, K, \bar{P}(M)) - S^*(Q, W, V, \dots)] dQ$$

where  $S^*$  reflects the additional cost incurred by affected firms in reducing risk by one "unit".





It is quite difficult to gather observations on the marginal rate of substitution between specific goods and risk reduction. Studies such as those by Burness, et. al. (1983) have attempted to elicit a value for risk reduction based on contingent valuation surveys. Alternatively, a number of studies of the wage premiums associated with risky occupations have focused on the willingness-to-accept compensation for assuming increased levels of risk on the job. We have chosen to focus on the latter group. This choice is motivated by two factors. The first is that the wage-risk premium studies reflect actual behavior with respect to risk. The second is that contingent valuation surveys may be significantly biased with respect to actual behavior (Bishop, Heberlein, and Kealy, 1983). In the context of the theoretical model developed above, use of the wage premium studies as a reflection of willingness-to-pay for risk reduction requires an important assumption. The assumption concerns the relationship between willingness-to-pay (WTP) and willingness-to-accept compensation (WTA). With respect to this question, it has been argued elsewhere that WTP and WTA can be assumed equal subject to certain restrictions (Freeman, 1979). In fact, most studies that discuss the value of risk in the labor market treat WTP and WTA as equal (Dillingham, 1984; Violette and Chestnut, 1983). To the extent that WTP and WTA are equal and wages foregone can be thought of as representing a decrease in consumption, we then have a good proxy for  $p_p \cdot \frac{U_{\bar{P}}(M)}{U_Q}$  and hence a measure of  $p_p$ .

### III: Previous Studies and Their Applicability

Considerable work has been done on the valuation of the wage premium required to induce workers to accept an increased risk of work-related



fatalities. These analyses are summarized in Dillingham (1984), and Violette and Chestnut (1983). Utilizing data on the wage differentials accruing to workers in different industries with varying work-related fatality rates, these studies generate estimates of the additional amount of wages an individual, or more commonly a group of individuals, will require to accept an increased probability of a work-related fatality.

Dillingham (1984), considered seven different estimates of the value of a statistical human life. Noting the considerable variation in the value-of-life estimates, ranging from \$330,000 to \$5,390,000 (1979 dollars), he tested to determine whether the definition of the risk variable in the different estimates could be assumed to account for at least part of this variation. Dillingham concluded that the risk variable definition does appear to have an influence on value-of-risk reduction estimates. Applying a variety of risk variable definitions to a common set of data on human-capital and control variables yielded value-of-life estimates ranging from \$1,375,000 to \$3,844,000.<sup>6</sup> However, while there is still considerable variation in the value of life estimates generated, this range is much reduced from that associated with earlier studies, and is used as the basis for the remainder of our analysis.<sup>7</sup>

Based upon Dillingham's estimates, a reduction in the probability of mortality of  $(1 \times 10^{-6})$  has an estimated value ranging from \$1.38 to \$3.84 per person (Found by multiplying \$1,375,000 and \$3,844,000 by  $(1 \times 10^{-6})$  respectively). Because the value-of-life estimates are based on annual mortality rates, the value of a reduction in the probability of mortality can be interpreted as an annual willingness-to-pay for a reduction in the probability of mortality. This latter point suggests a rough corres-



pondence between the figures cited here and the valuation of reductions in the probability of mortality in the near term, i. e., within one year, as discussed in Part II above.

While the preceding discussion has focused on a reduction in the probability of mortality of  $(1 \times 10^{-6})$ , the value of smaller or larger reduction in this probability can be similarly estimated. These values can then be combined with estimates of the relevant population size to determine the benefits associated with specific policies. Depending upon the reduction in the probability of death realized via the policy in question and the estimated size of the relevant population, the lower- and upper-bound estimates of the value of a given reduction in the probability of immediate mortality are calculated as:

$$(9a) \quad \text{LBE} = \$1,375,000 \times \text{AP} \times Z$$

$$(9b) \quad \text{UBE} = \$3,844,447 \times \text{AP} \times Z$$

where LBE and UBE are the lower- and upper-bound estimates respectively, AP is the size of the affected population, and Z is the reduction in the probability of mortality in the near term.

Use of Equations (9a) and (9b) for the value of risk reductions to be considered here requires an understanding of the biases that may be introduced through their use.<sup>8</sup> The biases considered include; 1) voluntary versus involuntary risks, 2) the relationship of the probability of mortality to low-level environmental risks in general, and 3) the effects of extrap-



olating values of risk reduction from the estimates in the studies considered.<sup>9</sup>

### 1) Voluntary Versus Involuntary Risks

Low-level environmental risks are usually the result of activities that are beyond the control of the average individual. As such, any risk that the individual is subject to as a result of such activities is borne involuntarily to the extent that the individual cannot force the cessation of the risk-generating activity. There may be cases where the individual has chosen to put himself in close proximity to the risk in question. However, it is assumed here that such instances are rare and that environmental risks are, for the most part, involuntarily assumed. This is in contrast to the studies which focus on wage differentials for jobs with varying levels of risk, assumed to be borne voluntarily, as a means of deriving estimates of the willingness-to-pay for reductions in risk. The obvious question is whether individuals can be assumed to be willing to pay the same amount for an equal reduction in a voluntarily-assumed risk and an involuntarily-assumed risk.

Previous analyses suggest that willingness-to-pay for risk reduction is context dependent. As an indication of the possible bias in the case of voluntarily- versus involuntarily-assumed risks, consider the following example noted in Weinstein and Quinn (1983). Parents have been overwhelmingly in favor of paying large sums of money to remove asbestos from school buildings, even where ambient levels are below the threshold of detection. On the other hand, seat belts and child restraints in cars go relatively unused despite the minimal cost involved in using them and the significant





potential for lives saved. This example clearly demonstrates a difference in willingness-to-pay for an involuntary- versus a voluntarily-assumed risk.

In another study by Starr (1969), it was concluded that the public appears to be willing to accept voluntarily-assumed risks that are a thousand times greater than involuntarily-assumed risks yielding the same benefit. The implication once again is that the willingness-to-pay for reductions in involuntarily-assumed risks is greater than the willingness-to-pay for equal reductions in those risks voluntarily assumed.

Finally, Fischhoff et al. (1979) analyzed a variety of risks and associated benefits in the context of Starr's analysis and concluded once again that there is an apparent tendency for individuals to value reductions in involuntarily-assumed risks more highly than reductions in those risks voluntarily-assumed. This constitutes the first source of downward bias in the estimates considered above because the estimates are, presumably, for voluntarily-assumed risks.

## 2) Changes in the Probability of Mortality versus Changes in Low-Level Environmental Risks

The second potential source of bias in the wage-risk premium estimates concerns the actual reduction in risk realized when a particular environmental hazard is reduced by some amount. It is reasonable to assume that in most instances, reducing the level of a specific environmental risk will result in a reduction in more than simply the probability of mortality.

Many environmental hazards result in significant environmental degradation, e.g., air and water pollution in excess of assimilative capacities.



These pollution activities in turn pose a substantial threat to the well being of ecosystems and can lead to morbidity in addition to mortality in humans. For instance, exclusion of a known carcinogen from a hazardous waste landfill will entail a reduction in, at a minimum, the probabilities of environmental damage via the reduced threat of groundwater contamination, and mortality and morbidity. The implication is that to the extent that the estimated wage-risk premiums are an accurate measure of the value of specific changes in the probability of mortality, they will nonetheless understate the total value of the reduction in most environmental risks. This constitutes a second major downward bias in the estimates considered.<sup>10</sup>

### 3) Extrapolating From Wage-Risk Premiums

The third potential bias concerns the validity of using linear extrapolations to attach values to reductions in risk that differ from those associated with job-related fatalities. The probability of mortality considered in wage-risk premium studies assumes a value ranging from  $(1 \times 10^{-4})$  to  $(1 \times 10^{-5})$  (Dillingham, 1984). On the other hand, although there are no firm estimates, the probability of mortality associated with most low-level environmental risks is usually believed to be much smaller; on the order of  $(1 \times 10^{-6})$  to  $(1 \times 10^{-12})$  or less. This disparity in risk levels brings into question the relationship between the value of equal reductions in risks that assume different initial values.

Assume that we are faced with two levels of risk, e.g., probabilities of mortality, A and B, and that  $A > B$ . Intuitively, it would seem likely that the value of a given reduction in A ( $R^A$ ) would be greater than the value of the same reduction in B ( $R^B$ ). That is:



$$(10) \quad RA > RB$$

This is one of the theoretical conclusions of an analysis by Weinstein, Shepard, and Pliskin (1980). However, research by Raiffa (1969) and others (Weinstein and Quinn, 1983; and Schoemaker, 1982) suggests that, in fact,  $RA < RB$ .<sup>11</sup> If the latter conclusion is correct, and there is empirical evidence to support it, then linear extrapolations from larger to smaller initial risk levels may also lead to a downward biased value of a reduction in risk.

To summarize, the three potential biases discussed above all suggest that the use of estimated wage-risk premiums will undervalue reductions in low-level environmental risks. Consequently, assuming that costs are accurately estimated, use of those estimates will provide a lower bound on the net benefits that a given policy can be expected to generate. As more data on the additional benefits, i.e., reductions in non-fatal risks resulting from the policy, is developed, this information could be combined with the wage-risk premiums to obtain a more accurate measure of the total benefits realized.

#### IV: An Application of the Process

As a means of demonstrating the approach outlined here, consider the following example. In a recent study, Cuddihy, Griffith, and McClellan (1984) examined the potential effects of a partial conversion of the light-duty vehicle fleet in the U.S. from gas- to diesel-powered engines. The shift to diesel-powered vehicles was assumed to be a likely response to the demand for increased fuel efficiency by both consumers and government





regulation. The primary purpose of the study was to assess the potential health effects associated with an increase in ambient particulate levels due to the increased diesel exhaust. The conclusion was that in addition to generating substantial savings in fuel expenditures, such a changeover would result in an increased number of lung cancer-related deaths in the U.S. The analysis was based on a number of assumptions. First, it was assumed that over roughly a twenty year period the number of diesel-powered light-duty vehicles will increase from 2% of the total fleet to approximately 20%. Second, it was assumed that the total light-duty fleet size will number approximately 165,000,000 vehicles at the end of the same time period implying that roughly 33,000,000 will be diesel-powered. Finally, it was assumed that diesel-powered vehicles will average approximately 20% greater fuel efficiency than similar gas-powered vehicles.

In order to assess the net impacts of the projected shift to diesels, assume that a rule is proposed that would ban the shift from gas- to diesel-powered vehicles. Consider first the benefits of the suggested rule. Based on their assumptions, Cuddihy, et al., estimated that approximately 200 additional lung cancer related deaths per year in the U.S. could be attributed to the projected increase in the use of diesel-powered light-duty vehicles. On average, and assuming a U.S. population of approximately 235 million people, this increase in the lung cancer mortality rate translates into an average increase in the probability of mortality from lung cancer of  $(8 \times 10^{-7})$ .<sup>12</sup> Based on the estimates discussed in Part III above, it is estimated that, in the aggregate, individuals would be willing to pay approximately \$275 million to \$768.9 million per year (in 1979 dollars) to avoid this increase in the probability of lung cancer-related mortality.



(The lower and upper bounds of this range are calculated via (9a) and (9b) respectively.) Recall that this range only takes into account the benefits of reducing the probability of mortality and ignores any other health and environmental quality related benefits from the rule which might accrue.

The primary costs of the rule prohibiting the shift from gas- to diesel-powered vehicles consist of the reduction in fuel expenditures foregone adjusted for increased maintenance and purchase costs. Consider Figure 1. The demand curve (D) represents aggregate demand for transportation (miles driven) and for simplicity is assumed to be perfectly inelastic over the relevant range. The supply curve labeled  $S_1$  reflects the marginal cost of miles driven. Assuming that fuel, and per mile maintenance and purchase costs are constant, the supply curve is perfectly elastic as indicated in Figure 1. Let  $S_1$  be the supply curve for miles driven before the shift from gas to diesel-powered vehicles. If the switch to diesels is a rational choice, the new supply curve for travel would lie below  $S_1$  such as at  $S_2$  reflecting lower cost per mile traveled due to the increased fuel efficiency of diesel-powered vehicles. The economic gain of the shift from  $S_1$  to  $S_2$  is measured by the area (a) and represents an increase in consumer surplus in the market for transportation. Conversely, the area (a) measures the cost of not allowing the shift from gas to diesel powered vehicles.

In 1982 a study of the economic and health impacts of dieselization of the U.S. light-duty vehicle fleet was performed by the National Research Council (NRC, 1982). Using the NRC study as a base and the assumptions of Cuddihy, et al., we can estimate the costs of our hypothetical rule as



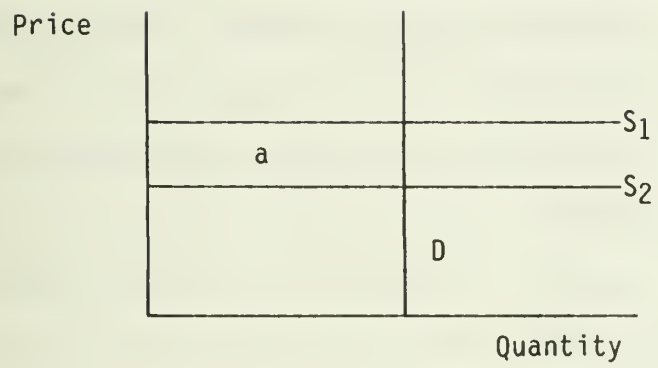


Figure 1. Demand and Supply for Transportation (miles travelled)



follows. First, using the NRC's estimated average fuel efficiency for medium sized gas-powered vehicles (assumed here to be an average of all vehicles), the 20% increase in fuel efficiency for diesels assumed by Cuddihy, et al., and assuming a constant real price of fuel (gas and diesel) of \$1.25 per gallon we estimate that vehicle owners switching to diesel would each save approximately \$457.22 over a ten year period (discounted 10%, 1980 dollars). Assuming 33,000,000 diesels, this translates to reduced fuel expenditures of approximately \$15,088 million. Next, assuming that additional maintenance expenses are approximately \$20 to \$40 per vehicle per year for 10 years, total additional discounted maintenance costs would range between \$4,055 million and \$8,111 million. Finally, assuming that the price of a diesel vehicle is \$300 to \$600 greater than that for a similar gas-powered vehicle, individuals would incur additional expenses of \$6,038 million to \$12,166 million. Overall then, the total savings if consumers substituted diesel-powered for gas-powered vehicles for 20% of the light-duty fleet, would range between approximately -\$5,189 million and \$5,006 million.<sup>13</sup> Conversely, this represents the cost of not allowing the substitution to take place.

In order to compare the benefits and costs of our hypothetical rule, we must first adjust the estimated annual benefits to 1980 dollars. Based on the consumer price index, the estimated annual benefits of 200 lung cancer related deaths range from \$312 million to \$872.7 million per year. Assuming a 10% discount rate, the present value of such benefits accrued over 10 years would then range between \$1,917 million and \$ 5,362.4 million.

In those cases where the application of the approach considered here yields a lower-bound estimate of benefits that exceeds the upper-bound





estimate of costs, further study of the rule would not be necessary. Based upon the figures presented here, it is not possible to conclude with certainty whether the net benefits of our proposed rule are indeed positive. However, the figures do suggest that further examination of the rule would be warranted. Moreover, we feel that this example is a good demonstration of the potential usefulness of the approach to benefits estimation being considered here.

#### IV: Conclusions

In this paper we have considered the potential for using previous studies of wage-risk premiums to attach values to the potential benefits of policies designed to reduce low-level environmental risks. The need for considering such an approach derives from the increasing demand for supporting economic analyses of the expected impacts (benefits and costs) of proposed government regulations. Our conclusion is that the use of these estimated wage-risk premiums holds considerable promise as a viable first step in assessing the benefits of policies designed to reduce the level of specific environmental risks. This conclusion rests on three points; 1) the immediacy of the situation, 2) the relationship of the wage-risk premiums to a theoretical treatment of the problem, and 3) the generally downward biased nature of the estimated benefits generated via this approach. These three points are considered in order.

First, policy makers are currently faced with the dilemma of enacting regulations in a relatively short period of time while simultaneously ensuring that the benefits of such rules can be expected to exceed the costs incurred. This is especially true in areas such as hazardous waste



disposal. However, there is scant information on how the public values policies such as these, and as a consequence a proxy for the benefits of such policies is needed.

Second, according to the theoretical model developed in Part II, individuals' valuations of reductions in risk can be thought of in terms of the amount of other goods they are willing to forego. Wage-risk premiums can be thought of as a proxy for this measure. To the extent that wages foregone to reduce risk leads to a decrease in consumption of goods and services, the measures are similar.

In Part III it was shown that wage-risk premiums are, by and large, downward biased estimates of the actual value of the benefits associated with reductions in low-level environmental risks. This implies that cost-benefit analyses utilizing these estimates can be expected to be biased in the direction of underestimating the actual level of net benefits realized. Hence the error of a false positive is minimized. In those instance where costs are estimated to exceed benefits by a small amount, the decision maker will be faced with the task of determining the likelihood that the additional benefits that can be expected to obtain would lead to benefits greater than costs. The fact that the estimates being used are downward biased could help guide their decisions in these situations as well.

The practice being suggested here is intended as a means of providing short-run input into the analysis of policies where the benefits of risk reduction is an important consideration. We feel that it could be especially useful in those instances where a number of different policy alternatives to a particular problem are being considered. Utilizing these numbers,



those alternatives that offer the greatest potential in terms of net benefits could be selected for additional, more costly analysis.

### Endnotes

1. This approach was also considered by the EPA in the development of its guidelines for completing regulatory impact analyses (Fisher, 1984).

While the intent was to use these figures to approximate the benefits of reductions in the probability of mortality the concept was dropped in the final version of the guidelines. This approach is also considered in Violette and Chestnut (1983), and Ehrenberg and Smith (1985).

2. Note that (1) could easily be modified to accomodate any particular environmental risk.

3. While this treatment of risk reduction as a purchasable good is not strictly applicable to all possible situations, we feel that it is justified to the extent that additional reductions in risk require the sacrifice of other goods and services (including public goods). This approach is also employed in Freeman (1979).

4.  $P(M)$  used here is defined as in Equation (1). In the present model one could think of  $P(M)$  as a function of the aggregate amount of good  $Q$ ,  $\bar{Q}$ , actually purchased. Since price taking behavior is implicitly assumed, the individual does not percieve that they can impact  $\bar{Q}$ .



5. The value of the incremental reduction is arbitrary and may be set at any level, say for example  $(1 \times 10^{-6})$ . However, if the purpose of the analysis is to compare the net benefits of different policy alternatives, once an increment is selected it must be held constant across the various policy options being considered in order to make valid comparisons.
6. As indicated in the text, Dillingham was interested in the effect of the risk-variable definition on the value of the estimated wage-risk premium. The data on the human-capital and control variables used by Dillingham were taken from the 1977 Quality of Employment Survey conducted by the Survey Research Center at the University of Michigan. The risk measures are based on data from the U.S. Department of Labor or were constructed by Dillingham in earlier work. Finally, most of the risk measures have been used in other wage-risk premium studies (Dillingham, 1984).
7. The justification for the use of this range rests on the assumption that the data used by Dillingham (1984) can be considered to be as good or better than that employed in similar studies. As noted in Dillingham, the data in the 1977 Quality of Employment Survey contained new and improved information about work and specific workers. As noted in Smith (1979) many of the other studies were forced to use data sets constructed by matching average job characteristics to individuals. Smith notes that this introduces an errors-in-variables problem that tends to bias tests against finding wage-risk premiums.
8. Violette and Chestnut (1983) also noted a number of these biases.





9. A fourth potential bias concerns the degree to which the wage-risk premiums considered here reflect only the value of the probability of mortality. This potential bias is cited in Violette and Chestnut (1983) where it is noted specification error in the equations used to estimate the wage-risk premiums could lead to an over-estimate of the value of a given reduction in the probability of mortality. However, to date there is no empirical evidence to support or refute this possibility.

10. We should stress that we are not suggesting that the estimation of the benefits of reductions in non-fatal risks should be ignored. Rather, the methodology suggested here is meant to serve as a "first-pass" approach that could eventually be supplemented by estimates of the benefits of reductions in the associated non-fatal risks. It also needs to be noted that the specification error referred to in note 9 above could also have an influence.

11. One of the reasons for this possibility offered in the studies cited concerns what is referred to as the "certainty effect". The argument is that people may value a given reduction in risk more highly when the reduction is more certain to result in a favorable outcome, e.g., when the initial level of risk is low to begin with.

12. The distributional impacts of the increase in the probability of mortality would fall disproportionately on the U.S. population with the largest impact falling on workers in urban expressways and street canyons. However,



treating the effect as equal across the population does not affect the value of the estimated benefits derived below.

13. The lower bound of -\$5,189 million in savings is unrealistic in that consumers would not be acting rationally if they did shift to 20% diesels. A more realistic lower bound on the range of potential savings would be \$0.

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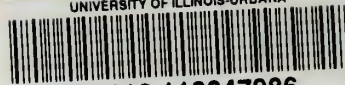
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